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16. ABSTRACT

The purpose of this paper is to summarize know effects of acid precipitation on stream ecosystems, with emphasis on the Great Lakes region. Unfortunately, very little research has been conducted on streams in much of the Great Lakes region. Thus, data will be utilized from other regions of the United States as well as from Norway, Sweden, and elsewhere; from laboratory studies on effects of pH on biota; and from our own research on effects of pH on soft water streams in Michigan to summarized probable effects on stream ecosystems within the region.

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CHAPTER 9

THE EFFECTS OF ACIDIFICATION ON STREAM ECOSYSTEMS

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INTRODUCTION

A previous paper in this conference by O.L. Loucks (see Chapter 2) has delineated the acid precipitation problem on a worldwide basis with emphasis on the Great Lakes region of North America. The purpose of this paper is to summarize known effects of acid precipitation on stream ecosystems, again with emphasis on the Great Lakes region. Unfortunately, very little research has been conducted on streams in much of the Great Lakes region. Thus, data will be utilized from other regions of the United States as well as from Norway, Sweden, and elsewhere; from laboratory studies on effects of pH on biota; and from our own research on effects of pH on soft water streams in Michigan to summarize probable effects on stream ecosystems within the region. Data from studies of acid mine drainage will be used to some extent. However, these data will be used with caution, since many of the effects of acid mine drainage on biota are likely the result of toxic levels of heavy metals and are not comparable to acid precipitation effects. Thus, much of this summary will rely on the limited data available on acid precipitation effects. All components of the stream ecosystem will be emphasized except fish, which are covered in detail elsewhere by Schofield (1976).

This chapter consists of a general review of the literature followed by a description of preliminary results from our stream research.

Acid Precipitation: Effects on
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EXTENT OF THE PROBLEM IN THE GREAT LAKES REGION

The sensitivity of streams to acid precipitation is coupled closely with the type of soil, the thickness of the soil, and bedrock geology of the regions they drain. Basically, this sensitivity is linked to the buffering capacity of the soils and waters of the watershed-stream ecosystem. In general, the higher the calcium carbonate content and the finer textured the soil, the less sensitive the soil is to acidification and the less likely it is that streams draining such watersheds will be acid. Streams draining areas of exposed, non-calcareous, granitic bedrock of low solubility and with thin soils of little buffering capacity, such as the Precambrian Shield area of Canada and some of the northern and mountainous portions of the United States, are especially sensitive to acid precipitation. The sensitive areas of the western Great Lakes region in the U.S. are concentrated in northern Wisconsin, northeastern Minnesota including the Boundary Waters Canoe Area, and the western portion of the upper peninsula of Michigan (Galloway and Cowling, 1978; McFee, 1980). Within this region of young, coarse textured glacial deposits, highly sensitive areas are interspersed with regions with little sensitivity at all (McFee, 1980). The streams reflect this, with soft water streams often being in close proximity to relatively hard water streams (Zimmerman, 1968).

Much of the western portion of the Great Lakes drainage in Ontario, with the exception of a few small areas of calcareous bedrock, is sensitive to acid precipitation, with the sensitivity extending to the boundary of the Precambrian Shield just south of the Muskoka-Haliburton region of Ontario south and east of the southern end of Georgian Bay (Dillon et al., 1978; Galloway and Cowling, 1978; Scheider et al., 1979). The carbonate regions south of this area in Ontario are not sensitive to acid precipitation inputs.

The sensitive areas of New York are concentrated in the Adirondack region or further east, primarily east of the Great Lakes drainage (McFee, 1980). Likewise, the Lake Erie dainage of Pennsylvania is not sensitive to acid rain inputs, with the most sensitive regions being in the Alleghany uplands and ridges further to the east (Arnold et al., 1980; McFee, 1980). With the exception of slight sensitivity of portions of southern Ohio and Indiana, Ohio, Indiana, Illinois, the lower peninsula of Michigan, and southern Wisconsin are not sensitive to acid precipitation (Galloway and Cowling, 1978; McFee, 1980).

The general statement that an overall region has soils and waters with high buffering capacity and low sensitivity to acid precipitation should not be taken to mean that no problem streams can exist in these areas. For example, streams on

isolated areas of coarse textured soils or exposed bedrock or in sandy soils near the Great Lakes can have low alkalinities and little buffering capacity. However, such streams will be rare and, in general, the areas outlined above are most likely to contain acid-sensitive streams.

The size of the stream also influences its sensitivity to acid precipitation inputs. Even in areas of igneous bedrock, larger streams are essentially not affected by acid precipitation because of a two-step neutralization process (Johnson, 1979). The first step of this process consists of neutralization of acid rain in the upper soil by dissolution of pre-existing aluminum hydroxide compounds and by leaching of bases from biologic matter so that rain has already been neutralized to a great extent before it reaches the stream. This neutralization is accompanied by high leaching losses of aluminum to the stream (Cronan and Schofield, 1979; Johnson, 1979). As this water moves downstream, the second step occurs. This step consists of chemical weathering reactions with loss of aluminum and gain of strong bases and silica by the water. Carbonic acid replaces the strong acids (sulfuric and nitric) as the major acid present even though total alkalinity is still very low. These changes seem to occur as a consequence of stream order regardless of elevation, with major streams not being susceptible to acidification (Johnson, 1979). Neutralization occurs rapidly in even the smallest streams in carbonate areas. Fairly large streams can be sensitive if much of the basin is exposed granitic bedrock. One of the largest streams known to have been affected is the Tovdal River in Norway with peak discharge in excess of 300 m^3/sec (Overrein et al., 1980). The effects of acid precipitation on streams of the Great Lakes region will probably be significant only in small, low-order (unbranched or with only one or two branches), headwater streams in areas draining thin, coarse-textured, non-calcareous soils or in areas of exposed hard-to-weather silicate bedrock such as parts of northern Wisconsin, northeastern Minnesota, the western portion of the upper peninsula of Michigan, and most of Ontario.

Fortunately, many of the western areas of the Great Lakes are not yet receiving rain as high in acidity as are areas further east. Thus, the most sensitive areas of the western Great Lakes region in the United States are receiving the least inputs. Even so, increased construction of power plants to burn western coal are likely to increase exposure of this region to acid rain during the next few years. South central Ontario already has a significant problem with acidification of streams (Dillon et al., 1978; Jeffries et al., 1979; Scheider et al., 1979) as does much of the high-altitude areas east and south of the Great Lakes in the eastern United States.

EFFECTS ON STREAM CHEMISTRY

Acid precipitation contains highly elevated levels of hydrogen ions, ammonium, nitrate, and sulfate, and decreased levels of bicarbonate compared to unpolluted rain. For the eastern United States, about 70 percent of increased acidity is sulfuric acid while about 30 percent is nitric acid (Likens et al., 1979). These chemical constituents are highly modified by watershed processes with combined watershed-stream interactions resulting in major changes in stream chemistry. These changes in streams draining watersheds with low buffering capacity include increased levels of aluminum, manganese, calcium, magnesium, and potassium in streamwater and can include short-term increases in sodium as a result of replacement of these elements in the soil by hydrogen ions as part of cation exchange processes and weathering reactions (Cronan and Schofield, 1979; Hall et al., 1980; Gjessing et al., 1976; Overrein et al., 1980; Scheider et al., 1979). Increases in aluminum are especially important, with concentrations reaching toxic levels (>200 ug/l) for many aquatic species as acidification results in replacement of aluminum by hydrogen ions and also increases solubility of aluminum in water (Cronan and Schofield, 1979; Hall et al., 1980; Gjessing et al., 1976; Overrein et al., 1980).

The two major anions received from acid precipitation are sulfate (about 64%) and nitrate (about 29%)(Likens et al., 1979). These anions are received both as wet and dry deposition (rain, snow, dustfall, aerosol impacts, etc.). Most of the sulfate is not retained by the watershed and ultimately is leached to streams (Overrein et al., 1980). Therefore, elevated sulfate levels are characteristics of acidified streams with sulfate replacing the bicarbonate-carbonate complex as the major anion present. Leaching of sulfate is directly correlated to aluminum plus calcium plus magnesium losses, and Norwegian scientists have developed predictive equations for this relationship (Gjessing et al., 1976; Overrein et al., 1980).

Ammonium and nitrate-nitrogen inputs are usually retained by the watershed since nitrogen is often in short supply for terrestrial ecosystems. Thus, nitrate (the easily leached form of nitrogen) concentrations in acidified streams are not normally elevated over background levels for non-acidified streams. There is a normal cycle of increased nitrate in streams during the winter since plant and microbial uptake within the watershed is at a minimum. It is possible, therefore, that snowmelt events associated with mid-winter thaws could result in short-term elevations of nitrate in stream water. However, we know of no reports of such elevated levels.

In addition to aluminum, manganese, and the major nutrients described above, other metals may also increase in stream water as a consequence of acidification. The solubility of many metals increases substantially as pH is lowered from alkaline to acid conditions. This increased solubility should result in substantial increases in many of these metals in acidified stream water. For example, metals solubilized from average crustal rock in water should increase from 0.03 to 500 µg/l for iron as pH is reduced from 8.1 to 4.5, from 0.0003 to 25 µg/l for copper, from 0.006 to 170 µg/l for nickel, and from 0.004 to 80 µg/l for zinc (Kramer, 1976). Zinc and lead concentrations are elevated in acidified lakes in Norway (Overrein et al., 1980), but these increases could either be the result of increased deposition or increased leaching. Increased atmospheric deposition rates during the past 30 yr of silver, gold, cadmium, copper, lead, antimony, vanadium, and zinc were described for lakes in New York and New Hampshire by Galloway and Likens (1979). Many of these same elements plus arsenic and selenium were also elevated in deposition in the Norwegian studies (Overrein et al., 1980). Whether from leaching or from increased deposition rates, trace metals may be elevated in streams receiving acid precipitation.

Experimental acidification of Norris Brook, a third-order stream in New Hampshire, resulted in the increases in aluminum, manganese, calcium, magnesium, and potassium already discussed; in decreases or increases in iron and cadmium depending on time of sampling, but no significant change in copper, nickel, zinc, or lead (Hall et al., 1980). This experiment consisted of acidification to pH 4 of the stream channel only and may not reflect changes that would have occurred if the entire watershed had been acidified. Increased manganese levels as well as aluminum are well documented. Scheider et al. (1979) reported increases of manganese from 3 to >200 µg/l. They also reported elevated levels of mercury in fish tissue from acidified lakes. Thus, elevated trace metal levels other than aluminum may contribute to biotic stress in acidified streams.

Many of the soft water streams in the western Great Lakes region are brown water streams draining swampy areas. These streams contain large amounts of organic matter which can alter the toxicity of aluminum and other metals due to complexation of these metals with the organic matter (Dickson, 1978; Driscoll et al., 1980; Patrick et al., 1981). Thus, results from other areas may not be totally germane to the western Great Lakes. More studies of the interaction between pH, trace metal toxicity, and organic content of the water are needed for prediction of effects in this region.

Increased aluminum concentrations can also result in increased flocculation of organic matter from the water column (Dickson, 1978). Thus, acid lakes are often highly transparent with little suspended sediment. This same process could result in less organic matter in stream water.

Episodic acidification events may be especially important in the response of biota to changes in stream chemistry resulting from acidification (Overrein et al., 1980). Changes in pH may be as great as one to two pH units following snowmelt (e.g., see review of Jeffries et al., 1979) or large storms and can result in loss of organisms even if the stream has a pH during most of the year that sensitive organisms can tolerate.

Changes in stream chemistry as a result of acid precipitation in poorly buffered streams can be summarized as follows. Changes will generally include elevated concentrations of aluminum, calcium, magnesium, potassium, and manganese and will often include elevated concentrations of zinc, lead, and other trace metals. Elevated aluminum is often toxic to sensitive species of invertebrates and fish. It also leads to increased sedimentation of organic matter, a primary food source for many stream organisms. Thus, acidification effects on stream chemistry other than just hydrogen ion changes may have significant effects on the biota. It is difficult to determine whether biotic changes resulting from acidification are linked to sensitivity to pH, to aluminum, or to some other chemical change. The overall effects of acidification on various trophic levels of the aquatic community will be described below.

EFFECTS ON PLANTS

The effects of acidification on plant communities in streams have not been extensively studied. Most of the work has come from studies conducted in Norway. These studies are summarized by Hendrey (1976), Leivestad et al. (1976), and Overrein et al. (1980). The Norwegian studies were initiated because of field observations that suggested that acidified streams often contained heavy growths of filamentous algae. To document this observation, Hendrey (1976) diverted flow from Ramse Brook, an already acidified stream (pH 4.3 to 5.5), through artificial stream channels and adjusted the pH to 4.0 in one channel, to 6.1 in another, and left it as ambient (4.3 to 5.5) for the third. It is important to note that ambient conditions were acid with acid-tolerant plants already present. Dominating species in all three channels were the filamentous green algae, Mougeotia sp., and the diatom, Tabellaria

flocculosa. These two species accounted for greater than 50 percent of total cells for all periods of study and usually for more than 70 percent of total cells. At low pH, Mougeotia sp. was more common, while Tabellaria flocculosa was more common at pH 6.1. These are the same two species found to be dominant in most acidified streams of Norway. Acidification to pH 4.0 led to increased accumulation of algae, to increased chlorophyll levels, and to higher proportions of organic matter existing as algal cells. Carbon uptake per unit of chlorophyll was lowest at pH 4.0.

Hall et al. (1980) acidified Norris Brook, New Hampshire, from background levels of 5.7 to 6.4 to a pH of 3.9 to 4.5. This acidification resulted in 2- to 5.6-fold increases in chlorophyll-a and obvious visual increases in algal biomass in the stream. However, periphyton production rate was lower in the acidified stream despite the increased biomass.

Patrick et al. (1968) experimentally determined the effects of pH on diatom communities on glass slides in boxes with water diverted from Darby Creek, Pennsylvania, with a control pH of 7.3 to 10.1 and with a low pH of 4.7 to 5.7. They reported that low pH resulted in greatly reduced biomass with maximum reduction occurring in October and November and least reduction occurring in May and June. Likewise, community structure as measured by diversity was reduced more in October and November than in May and June by lowered pH, but overall species differences were not very great, with only a few species present at low pH that were not present in controls and vice versa.

Warner (1971) reported that sections of Roaring Creek, Virginia, polluted by acid mine drainage with a pH of 2.3 to 3.8, had fewer species of periphytic algae than did unpolluted sections of the stream (27 or more species in unpolluted sections, 10 to 19 in polluted sections). Dominant forms in acid waters included Ulothrix tennerima, Pinnularia termitina, Eunotia exigna, and Euglena mutabilis.

Muller (1980) conducted experiments on the effects of acidification on periphyton in lakes. Diatoms were dominant at pH levels above 6 but were replaced by green algae below 6 with Mougeotia becoming the dominant species at pH 4. Fewer species and lower species diversity were observed at lower pH. Standing crops increased at low pH, but production did not. Thus, results were very similar to results from acidification of streams.

While data are limited, benthic algal communities in streams appear to respond to acidification by:

(1) species shifts to acid-tolerant forms, often the filamentous green algae Mougeotia sp.;
(2) lower diversity;
(3) increased standing crop of plants overall, but decreased standing crop of many diatoms; and
(4) no change or lower production rates by plants.

Increased standing crop with same or reduced production rates is often attributed to one or a combination of the following:

(1) lower rates of decomposition,
(2) lower grazing rates by herbivores, and
(3) species shifts to a few acid-tolerant species (Hall et al., 1980; Hendrey, 1976).

There are few data on other types of plant communities in streams. In preliminary results of experiments at Monticello, Minnesota, acidification from pH 7.9-8.2 to 5.0 of outdoor, non-shaded artificial channels with low flow rates resulted in decreased diversity of macrophytes, invasion of terrestrial plants into the stream and changes in the stream from an autotrophic to a heterotrophic system, perhaps because of shading by the emergent plants and the expanded cover of duckweed (Lemna minor)(U.S. EPA, 1979). These results cannot readily be extrapolated to other systems, since the background levels of alkalinity would make this type of stream fairly insensitive to acid rain inputs.

There are some data on responses of macrophytes to acidification in lakes that can give insight into possible effects in streams. In general, effects on macrophyte communities tend to be the result of indirect factors rather than actual intolerance to increased hydrogen ions (Hultberg and Grahn, 1976). These indirect effects include the existence of more free carbon dioxide as the lake becomes acid, which favors plants with high free carbon dioxide requirements such as many of the mosses. Sphagnum, the peat moss of bogs, forms mats on the bottom of many acidified Swedish lakes and has replaced Lobelia and Isoetes as a result of over-growing them. Likewise, decomposition rates in lakes are slower, and dense felt-like mats of algae and fungi form over the sediments of many acidified lakes. This mat competes with macrophytes for sediment nutrients, reduces flow of nutrients into overlying water, and shades out Lobelia and Isoetes. The reduced decomposition rate also results in increased organic content of the sediments. Increased organic content leads to increases in the acidophilic, high organic-requiring species, Juncus bulbosus var. fluitans. Other changes described as a consequence of acidificaton of Swedish lakes include decreases

in Phragmites communis and Myriophyllum sp. but no effects on Nymphaea alba or Nuphar luteum (Hultberg and Grahn, 1976).

Since many of these lake macrophytes also occur in and along streams, similar changes mght be expected in the stream macrophyte communities.

EFFECTS ON DECOMPOSITION

Acidification of streams leads to decreased rates of decomposition of organic matter (Leivestad et al., 1976; Overrein et al., 1980). Decreased rates of decomposition in small, acid-sensitive headwater streams can have major impact since most energy used by organisms in these streams is derived from organic matter which enters the streams from adjacent terrestrial ecosystems.

Experimental evidence for decreased rates of decomposition is somewhat limited and is derived primarily from experiments conducted in Norway (see summaries by Leivestad et al., 1976; Overrein et al., 1980). One set of these experiments was conducted on microbial activity on peptone in the laboratory. As pH decreased, numbers of species of ciliated protozoans and total bacterial cell counts decreased. Below pH 5, nitrification was reduced, oxidation of ammonia ceased, and decomposition rates were reduced. Below pH 4, numbers of fungi increased. In experiments with glucose and glutamic acid, lag time before start of rapid decomposition was increased at lower pH but decomposition rates after this lag were not appreciably affected down to pH 3.5, perhaps as a result of selection and adaptation of microbes. There was a shift to fungal populations as pH was lowered. Similar experiments with homogenized wilted birch (Betula verrucosa) leaves did show reduced decomposition rates during the three-week period of these studies in the laboratory.

Experiments were conducted on the effects of pH on decomposition rates of birch leaves using artificial channels and Ramse Brook water in Norway (Leivestad et al., 1976). At pH 4, decomposition after one year was lowest (about 46% weight loss) and was significantly different at the $P < 0.001$ level from decomposition at pH 6.0 (about 52% weight loss). Similar experiments with aspen wood chips yielded similar results as did experiments carried out in natural streams.

Hall et al. (1980) reported that acidification of a New Hampshire stream resulted in growth of a basidiomycete fungus on tree roots and mosses in the stream and resulted in a thin layer of fungal hyphae over 70 percent of the stream bottom.

Maple and birch leaves had similar genera of aquatic hyphomycetes as did those in non-acidified stream sections nearby but species diversity and spore numbers were reduced in the acidified section of stream.

Preliminary data from open artificial channels in Minnesota suggested that decomposition rate was slowed substantially by acidification (61% weight loss at pH 7.9-8.2 to 48% at pH 5.0) in litter bags open to detritivores while there was no significant difference when detritivores were excluded (29% weight loss at pH 7.9-8.2 versus 26% weight loss at pH 5.0). This study suggested that the microbial rates were not substantially reduced but that decomposition rates were reduced by reduction in species diversity and numbers of dominant macroinvertebrates (U.S. EPA, 1979).

Decreased decomposition, accumulation of organic matter on the bottom, and development of thick gelatinous mats of fungal hyphae on lake bottoms are known to occur widely as a result of acidification. These processes are also known to reduce remineralization rates of nutrients such as phosphorus and can lead to self-accelerating oligotrophication of such acid lakes (Dickson, 1978; Leivestad et al., 1976; Overrein et al., 1980). Such changes in stream chemistry could also lead to major losses of stream productivity since organic matter forms the major food resource for animals in small, headwater streams.

EFFECTS ON INVERTEBRATES

Acidification can have major impacts on the invertebrate fauna of a stream. Certain species appear to be much more sensitive than others. Thus, community structure is likely to be altered as sensitive species are replaced by more tolerant forms. This reorganization of community structure has not been studied in detail. Instead, studies have been conducted on the species level or on a macrodistribution level (stream surveys of acid versus non-acid streams). The studies of Hall et al. (1980) did examine differential drift rates among different functional groups of invertebrates as a consequence of acidification.

Surveys of streams to ascertain which organisms occur within a region in rivers with low versus high pH, which species are tolerant of lower pH, and which species appear to be favored by lower pH are useful in predicting sensitivity to acidification of streams. However, these data must be used with caution since many other factors are linked to low pH. For example, soft waters with low levels of calcium, magnesium,

sodium, and potassium can limit the distribution of certain species with high requirements for any of these elements. Snails and clams, for example, have a high requirement for calcium carbonate for shell building and may not exist in soft water. In fact, most snails cannot survive in streams that are more acid than pH 6.2, with most but not all species requiring more than 15 mg/l of carbonates (Pennak, 1978). Soft water is highly sensitive to acidification. Thus, absence of species may be correlated with low pH, but these species may be absent due to lack of calcium or carbonate for shell building or due to lack of the proper balance of calcium, sodium, and potassium ions necessary for osmotic regulation, etc. Thus, field survey work needs to be coupled to laboratory studies before cause and effect can be determined.

On the basis of distribution within the River Duddon in England, Sutcliffe and Carrick (1973) suggested that all mayflies (Ephemeroptera), certain species of caddisflies (Trichoptera) in the genera Wormaldia and Hydropsyche, the river limpet, Ancylus fluviatilis, and the amphipod, Gammarus pulex, were limited to waters with pH > 5.7, while stoneflies (Plecoptera), caddisflies in the genera Plectrocnemia and Rhyacophila and the family Limnephilidae, midges (Chironomidae), blackflies (Simulium) and craneflies (Tipulidae) were all present in acid waters (ph < 5.7) and appeared to be insensitive to pH. Minshall and Minshall (1978), while agreeing that there was a correlation with pH, disagreed that pH was the prime determinant of distribution in this river and suggested other water quality factors were more important.

The sensitivity of mayfly larvae and the relative insensitivity of stonefly larvae to low pH were also suggested by surveys of 25 stream sites in Norway (Leivestad et al., 1976). Sixteen of these sites had a pH between 4.0 and 5.0 and never contained more than 2 species (mean of 1.6) of mayflies. Above pH 5.0, numbers of species present varied between 1 and 7 (mean of 4.5) with all but one of the sites having at least 3 species present (Leivestad et al., 1976). Mean numbers of mayfly species are about 3 to 4 times higher in streams with a pH of 6.5 to 7.0 than in streams with a pH of 4.0 to 4.5 (Overrein et al., 1980). This relationship was not found for stream stoneflies (Plecoptera), even though it was found for lake stonefly species (Leivestad et al., 1976; Overrein et al., 1980).

Drift rates for mayflies increased as a result of experimental acidification (Hall et al., 1980; Herricks and Cairns, 1976-76; Overrein et al., 1980). Ephemerella and Epeorus showed largest drift densities as a result of acidification of a New Hampshire stream (Hall et al., 1980).

Drift rates of Baetis sp., Ephemerella sp., Isonychia sp., and Stenonema sp. increased substantially following a 15-minute shock treatment where pH of Mill Creek, a hardwater stream in Virginia, was reduced from pH 8.0 to 4.0 by addition of sulfuric acid. Drift remained elevated for 48 hr (Herricks and Cairns, 1974-76).

Other evidence for sensitivity of mayfly larvae (Ephemeroptera) includes laboratory studies of Ephemerella subvaria by Bell (1971) and Bell and Nebeker (1969). Their studies indicated that Ephemerella subvaria was the most sensitive of 10 insect species tested. Fifty percent of test species died after 30 days exposure to pH 5.38, while 50 percent successfully emerged at pH 5.9. However, not all members of the genus are as sensitive. Fiance (1978) found that experimental acidification of Norris Brook, New Hampshire, for six months from pH 5.85-6.45 to pH 3.9-4.3 with sulfuric acid did not affect emergence rates of Ephemerella funeralis but did cause a decrease in growth and nearly eliminated recruitment of the new cohort.

Other known acid-sensitive species of mayflies include Baetis rhodani, the most common species in rivers in Norway with a pH > 6.0. In rivers below pH 6, these species are eliminated. Experiments indicated that at pH of 4.5 to 4.7 and low conductivity, all species died in less than 48 hr while 10 percent survived the same pH with somewhat higher conductivity. Field observations indicated that Baetis rhodani does not reproduce and probably dies from physiological stress at pH below 5.0 (Overrein et al., 1980). Baetis sp. also had the highest drift rates as a result of a 15-minute shock treatment with pH reduced from pH 8 to 4 in Virginia (Herricks and Cairns, 1974-76).

The studies by Hall et al. (1980) suggested that stonefly larvae (Plecoptera) were relatively insensitive to low pH. Drift rates of the dominant species (Nemoura and Leuctra) did not increase during acidification. Malirekus/Isoperla did increase but numbers were too small to conclude much (from 0 to 8 individuals). Thus, the stream surveys in England and Norway, results from experimental acidification of streams, and laboratory tests (Bell, 1971; Bell and Nebeker, 1969) indicate that stoneflies (Plecoptera) in streams are fairly tolerant of low pH.

Stream insects are often grouped according to their ecological role in the stream into four functional trophic groups (Cummins, 1975). These groups are:

(1) grazers and scrapers--herbivores feeding on attached algae,

(2) shredders--large particle feeding detritivores,
(3) collectors--fine particle feeding detritivores which filter particles from the water or gather them from surfaces, and
(4) predators.

Hall et al. (1980) found that acidification of Norris Brook, New Hampshire, from pH 5.7-6.4 to pH 3.9-4.5 resulted in a 37 percent decrease in emergence of insects but that shredders and predators were unaffected, with all the decrease ascribed to collector organisms. These reductions in abundance were primarily due to decreased emergence of midges (Chironomidae, Subfamily Orthocladinae) and mayflies (Ephemeroptera). No scrapers emerged during the time of their study from either the control or acidified streams.

Hall et al. (1980) also studied the effects of acidification on drift of invertebrates. There was a 3.9-fold increase in overall macroinvertebrate drift on the first day of acidification followed by a peak 13-fold increase on the second day. Drift rates decreased to those typical of the control section one week after acid addition. There was a differential effect on functional groups. Drift of shredders was unaffected. Collectors were affected most with up to 17-fold increases in drift on the second day. Scrapers had a peak 9-fold increase on the first day, while drift of macroinvertebrate predators increased 4-fold.

Hall et al. (1980) also determined that there was a 75 percent reduction in total numbers of benthic invertebrates as a result of acidification. Aquatic Diptera formed about 68 percent of the numbers in the stream and were reduced by 87 percent by acidification. Cranefly larvae (Tipulidae) were reduced by 91 percent, biting midge larvae (Ceratopogonidae) by 88 percent, and the midge larvae (Chironomidae) by 86 percent. Chironomidae abundance in acidified lakes in Norway was also reduced by 60 to 80 percent (Overrein et al., 1980).

The 15-minute shock treatment of lowering pH from 8 to 4 in a hardwater stream in Virginia reduced total numbers of benthic invertebrates by 42 percent and reduced diversity by 28 percent. The stream took 28 days to recover from this stress (Herricks and Cairns, 1974-76).

Invertebrates other than insects are also known to be sensitive to low pH. Much of the data on these species come from lake studies in Norway. These lake studies give some insight into potential sensitivity of stream invertebrates. Amphipods (Gammarus) are known to be acid-sensitive in both lakes and streams. Gammarus lacustris does not tolerate pH

less than 6.0 in lakes in Norway, while Gammarus pulex is not found below pH 5.7 in the River Duddon in England (Overrein et al., 1980; Sutcliffe and Carrick, 1973).

Freshwater snail species start to be eliminated from lakes in Norway at pH 6.0 and never occur below pH 5.2. This sensitivity is to be expected in North America as well since snails rarely occur in streams below pH 6.2 (Pennak, 1978). Most small clams (Sphaeriidae) are eliminated below pH 6.0 with only 2 of 20 species occurring below pH 5.0 in Norway (Overrein et al., 1980). The two major North American families of bivalves (Unionidae and Sphaeriidae) are known to be acid-sensitive, with Unionidae rarely found below pH 7.0 and Sphaeriidae found at pH as low as 6.0 (Pennak, 1978). Thus, no bivalves are likely to occur in acidified streams.

Isopods (Asellus aquaticus) are less acid-sensitive in lakes and can tolerate pH levels as low as 4.8 (Overrein et al., 1980). Preliminary data from our investigations suggest Asellus intermedius will be eliminated from streams acidified to pH 4.0 (see results below).

Segmented worms (Oligochaetes) are also less abundant in acid lakes (3 to 4 times higher in non-acidified lakes) in Norway (Overrein et al., 1980) so reductions could occur in acidified streams.

The crayfish (Orconectes virilis) is able to survive pH 4.0 in lakes for short periods but eventually dies due to interference with uptake of calcium leading to an inability to successfully molt (Malley, 1980). This species is also widespread in streams, so acidification to pH below 5.0 could lead to its demise.

On the other hand, surface-dwelling organisms such as water-boatmen (Corixidae) do not seem to be affected by acidification, and become a dominant group in acidified lakes (Overrein et al., 1980). Stream surface-dwelling insects might also increase as a consequence of acidification.

While this coverage of effects on stream invertebrates is not all inclusive, it does indicate that major impacts on the invertebrate fauna are to be expected. Since these invertebrates are important to processing of organic matter in streams (Cummins, 1975) and as the basis of fish production (Brocksen et al., 1968), effects on them can lead to major changes in nutrient cycling rates, processing of organic matter, and fish production.

STUDIES OF POTENTIAL EFFECTS OF ACIDIFICATION IN MICHIGAN

Introduction

We are using artificial stream channels at the Kellogg Biological Station to study the potential effects of acid precipitation on soft water streams in Michigan. These studies have been underway for about 10 months but actual acidification has only been in progress for four months. The purpose of this part of the chapter is to briefly describe our studies and to report preliminary results.

Materials and Methods

Two small, first-order streams that drain directly into Whitefish Bay near Paradise, Michigan, were selected as a source of organisms for use in simulation. The alkalinity of these streams varies from 8 to 19 mg $CaCO_3$/l and hardness varies from 14 to 18 mg $CaCO_3$/l. They are slightly higher in nitrogen and phosphorus than some of the least perturbed streams of acid-sensitive regions such as those in the Huron Mountains of Marquette County, Michigan (Table 1).

The temperature- and light-controlled concrete channels used for simulation of these streams were described in detail by Cummins (1971)(Figure 1). These channels are 12 m long, 1.5 m wide, and vary from 0.6 m deep at the upstream reservoir to 0.8 m deep at the downstream reservoir. In actual operation, water in the channels is maintained at a depth of 14 to 20 cm just below the upstream reservoir to 31 to 37 cm just in front of the downstream reservoir. The deepening of the channel in a downstream direction results in simulation of a riffle and pool environment. The channels were cleaned thoroughly and repainted with epoxy paint prior to the start of these experiments.

The channels were lined with a mixture of the large gravel to stone sediments taken from the Peshekee River in the Huron Mountains, Marquette County, Michigan, and fine sand sediments typical of the streams that enter Whitefish Bay near Paradise, Chippewa County, Michigan. The sand forms a layer over the gravel and rock except in the faster current within 3 m of the upstream reservoir. The water source is deionized well water. This deionized water was allowed to equilibrate with the sediment and then adjusted to levels typical of upper peninsula soft water streams by adding appropriate amounts of various constituents. The stream water equilibrated with sediments so that few chemical additions were needed. Enough potassium

Table 1

Comparison of Chemistry of Soft Water Streams in Michigan Versus Artificial Channels Just Prior to Acidification

	Unnamed Streams[a]	Peshekee River[b]	Artificial Streams	
			Control	Acidified
	mg/l[c]			
Inorganic-nitrogen	0.16±.04	0.03	0.03	0.03
Organic-nitrogen	0.53±.09	0.04	0.07	0.12
Molybdate reactive phosphorus	0.038±.023	0.002	0.013	0.012
Total phosphorus	0.064±0.15	0.019	0.043	0.030
Chloride	1.75±.50	1	1	1
Sulfate	<1.0	2-3[b]	<1.0	<1.0
Hardness (as $CaCO_3$)	16.25±2.06	15	15	11
Alkalinity (as $CaCO_3$)	12.4±3.9	10-14[b]	17	13
pH (range)	6.2-6.8	7.3-7.7[b]	7.3	7.4
Calcium	5.24±1.22	4.8-6.0[b]	6.35	4.47
Magnesium	1.40±.45	1.0-1.4[b]	0.35	0.44
Sodium	0.80±.02	0.90-1.30[b]	0.63	1.40
Potassium	0.75±.26	0.30-0.40[b]	0.80	1.31
Aluminum (µg/l)	78±26	--	33	20
Manganese (µg/l)	6±7	--	1	1
Iron (µg/l)	287±146	400-500[b]	6	14
Copper (µg/l)	51±26	--	44	24

[a]Mean of samples from two unnamed streams near Paradise, Chippewa County, during October and November, 1980.

[b]Peshekee River at US 41 Bridge, Michigamme Township, Section 25, Marquette County, Michigan. Data marked by b are taken from STORET, other data collected in the field in June, 1980. Alkalinity varies from 10 to 14 mg/l for this site. Our data from further upstream indicate alkalinity as low as 3 mg $CaCO_3$/l.

[c]Except where indicated.

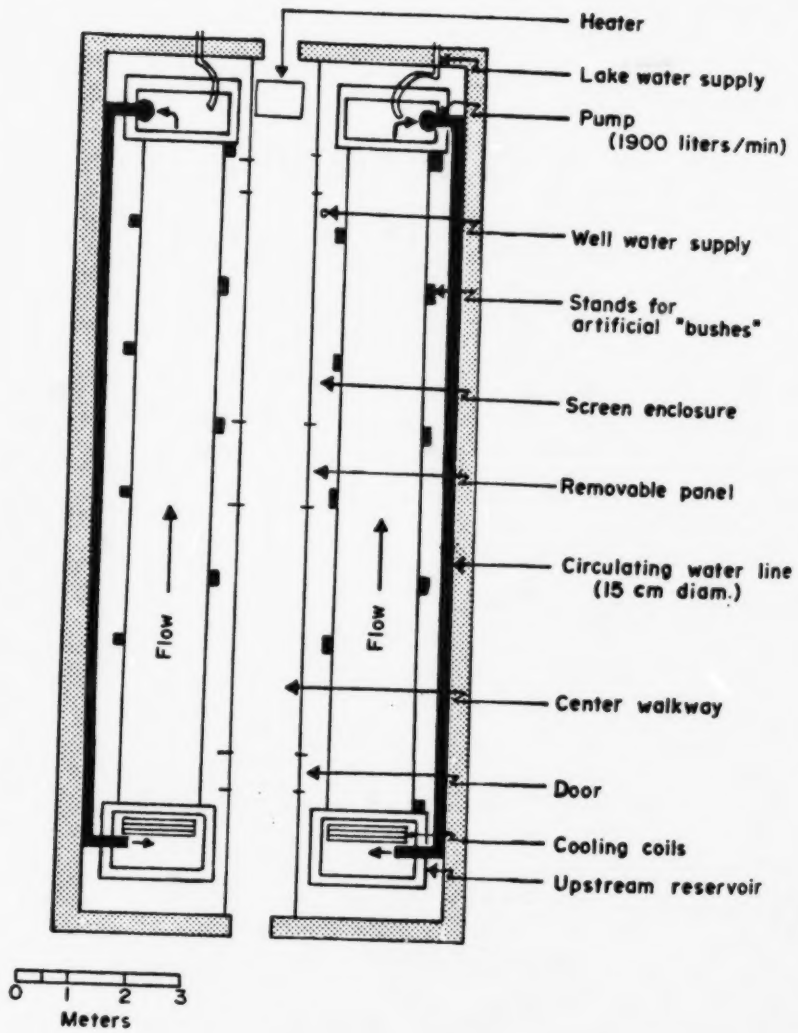

Figure 1. Diagrams of artificial stream channels used fo[r] simulation of soft water streams in Michigan (Cummings, 1971).

nitrate was added to add 0.10 mg NO_3-N/l and 0.28 mg K/l to each stream. Enough sodium monobasic phosphate was added to add 0.05 mg P/l and 0.04 mg Na/l to each stream. The nitrate and phosphate did not stay in solution long and were likely taken up by the biota and/or the sediments. Thus, levels in stream water just prior to acidification were somewhat low for the streams near Paradise but approximated concentrations typical of the Peshekee River (Table 1). Iron was very low compared to the natural streams and magnesium, aluminum, and manganese were somewhat low (Table 1). Otherwise, water chemistry appeared to reasonably simulate chemistry of upper peninsula streams.

Organisms were collected from the two small streams near Paradise, Chippewa County, Michigan, and were transported back to the laboratory in well moistened, chilled leaf packs. These organisms were used to stock the streams. Even though efforts were made to evenly divide these organisms, population levels for the two streams prior to experimentation were different. Sampling of loose leaf material deposited in the stream indicated that there were 14.5 $\pm$ 4.4 (mean $\pm$ 95% C.I.) organisms per 5 gm of leaf material in the acidified channel prior to acidification while there were only 6.3 $\pm$ 2.1 organisms per 5 gm of leaf material in the control stream. Sampling of leaf pack material prior to acidification yielded 19.5 $\pm$ 6.3 organisms per leaf pack in the channel that was to be acidified versus 9.0 $\pm$ 4.8 per leaf pack in the control. Thus, organisms appeared to be more concentrated in leaf packs than on the loose organic matter, but in each case there were 2.2 to 2.3 times more organisms per leaf pack in the stream that was to be acidified than there were in the control stream. These differences have to be taken into account in subsequent data analysis.

Five-gram leaf packs of white birch (Betula papyrifera) and sugar maple (Acer saccharum), the dominant tree species of the forest along the streams near Paradise, Michigan, were placed in the streams on the leading edge of bricks using techniques of Merritt et al., (1979). During initial stocking of the streams with invertebrates, these leaf packs were made up of leaves from terrestrial areas adjacent to the streams near Paradise, Michigan, since all leaf material in the stream had been decomposed by the start of the study in late May, 1980. Prior to acidification, part of these leaf packs were removed and all organisms collected and sorted to species while the other half were removed 38 days after acidification began. These "old" leaf packs were used, therefore, as a method of screening losses of invertebrates during initial acidification.

Newly fallen leaf material was collected in October, 1980, dried, and used to construct new leaf packs following the techniques of Merritt et al. (1979). Twenty 5 gm leaf packs

of each species (white birch and sugar maple) were placed in each stream on November 4, 1980. This was two weeks prior to removal of "old" leaf packs described above and 24 days prior to start of acidification. Since the start of acidification, these leaf packs have been removed from the stream, weighed, and all organisms identified and counted on a monthly basis. This process will continue until leaf material is degraded or until the experiment ceases in September, 1981.

Acidification began on November 28, 1980. The pH was dropped from 7.4 to 4.0 by slowly dripping 0.1 N sulfuric acid into the stream over a 6-hr period. The pH of 4.0 is maintained by dripping in appropriate amounts of acid on a daily basis. Thus, pH is maintained at levels between 4.0 and 4.3.

Some seepage of surface runoff through the high carbonate soils near the greenhouse enters the artificial streams despite our best efforts to stop it. Thus, hardness and alkalinity slowly increase over time and have to be reduced by deionization of stream water. Initial leaching of materials from sediments by acidification during the first 8 days increased hardness from 16 to 30 mg $CaCO_3$/l. The acidified stream is kept at this hardness (30 mg/l) by deionization of stream water. Alkalinity and hardness of the control stream are maintained between 15 and 20 mg $CaCO_3$/l, the level typical of both streams prior to acidification.

Preliminary Results

Acidification reduced the total number of benthic invertebrates on the "old" or conditioned leaf packs which had been in the stream for several months from 19.5 $\pm$ 6.3 to 7.2 $\pm$ 7.3 organisms per leaf pack or a 63 percent decease in total numbers after 38 days of acidification ($p < 0.05$, t test). Population levels in the control channel were not significantly different after 38 days (t test)(from 9.0 $\pm$ 4.8 to 7.8 $\pm$ 2.3 organisms per leaf pack) compared to the 63 percent reduction in the acidified stream over the same time period. The reduction in the acidified stream compares well to the 75 percent reduction in total numbers reported from acidification of Norris Brook, New Hampshire (Hall *et al.*, 1980). In fact, Hall *et al.* (1980) reported an 84 percent reduction in total numbers of invertebrates from debris dams.

The most common invertebrate present was a shredder, *Lepidostoma liba* (Trichoptera). It appears to be relatively unaffected by acidification. Numbers in the old leaf packs were reduced by 44 percent in the acidified stream from 8.8 $\pm$ 4.5 to 4.9 $\pm$ 6.2 organisms per leaf pack, but they were also reduced by 34 percent in the control stream from 3.5 $\pm$ 3.0 to

2.3 ± 1.0 organisms per leaf pack, but neither reduction was significantly different ($p > 0.05$, t test). Likewise, numbers on the "new" leaf packs (those put in the stream from newly fallen leaves in November, 1980) were reduced from 9.2 organisms per leaf pack based on collections of loose leaf debris prior to acidification to 7.8 after one month of acidification (a 15% reduction) while the control stream was reduced from 4.5 to 3.8 organisms per leaf pack (a 16% reduction; no significant difference in either case). Thus, natural mortality seems to be reducing population levels but no significant reduction can be ascribed to acidification effects. Both downstream drift rates and emergence of shredders were unaffected by acidification in New Hampshire as well (Hall et al., 1980).

The isopod, Asellus intermedius, is also an important component of the macroinvertebrate fauna in the natural streams near Paradise, Michigan. It was markedly reduced in numbers by acidification from 6.8 ± 4.5 to 0.9 ± 0.8 organisms per leaf pack on the "old" leaf packs after 38 days of acidification, or an 87 percent reduction in total numbers. At the same time, populations in the control stream increased slightly from 0.8 ± 0.7 to 0.9 ± 1.1 organisms per leaf pack. This same trend was characteristic of the new leaf pack with numbers reduced from 2.4 ± 1.2 per leaf pack to 0.25 ± 0.26 by acidification (a 90% reduction), while populations in the control stream expanded from 0.43 ± 0.28 organisms per leaf pack to 1.7 ± 0.61. Asellus aquaticus is eliminated from lakes in Norway at pH 4.8 to 5.2 (Overrein et al., 1980), and our findings suggest that A. intermedius in streams is also sensitive to acidification.

The snail Physa heterostropha had low population levels in the acidified stream prior to acidification and has apparently been eliminated as a result of acidification. Population of this species was larger in the control stream and population levels appear to have remained stable there (there was an apparent decrease from 2.8 ± 0.7 organisms per leaf pack to 0.9 ± 1.6 per leaf pack on the old leaf packs but an apparent increase from 0.4 to 1.5 organisms per leaf pack on the new leaf pack material).

Decomposition rates of leaf material have not been significantly affected by acidification after two months of exposure to pH 4.0. For example, sugar maple leaves retain 73.7 ± 5.0 percent of their original weight in the upper half of the acidified channel compared to 74.3 ± 1.1 percent in the upper half of the control channel and 78.2 ± 5.6 versus 74.7 ± 2.2 percent for the lower half of the acidified versus the control channel, respectively. Since decomposition rates are very low at low winter temperatures (Suberkropp et al.,

1975)(the streams have been at 2-3°C over the winter), differences due to acidification are not likely to show up until decomposition rates increase during spring. We expect these differences to occur based on studies summarized above, but they have not occurred as yet in our study.

The effects on stream chemistry due to leaching of material from the substrate by acidification can be summarized as follows. After 8 days of acidification, calcium increased 2.2-fold, magnesium increased 1.7-fold, potassium dropped slightly (13%), sodium increased slightly (1.3-fold), aluminum concentrations were over 6 times higher than pre-acidification levels, with copper, iron, zinc, and manganese all showing significant increases. The stream channel results may not be totally applicable to natural systems, and we are in the process of conducting leaching studies of stream sediments from the upper peninsula streams using acidified stream water in order to better predict these responses. As expected, sulfate levels increased due to addition of sulfuric acid, while carbonate-bicarbonate alkalinity was reduced to negligible levels.

The above summary represents preliminary results from our studies. We expect to continue these studies and to expand them to include individual growth, feeding rates, and mortality rates of Lepidostoma liba, Physa heterostropha, and Asellus intermedius as well as the collector, Parapsyche sp. under acidified and non-acidified conditions.

CONCLUSIONS

Acidification significantly alters the chemical and biological processes in soft water streams. These changes include increases in aluminum, manganese, calcium, magnesium, and potassium concentrations in stream water with aluminum concentrations increasing to potentially toxic levels.

Effects on the biota include increases in standing crop of benthic algae with a shift toward acid-tolerant filamentous green algae such as Mougeotia sp. However, productivity is often unaffected or may be reduced with the increased standing crop resulting from lower grazing pressure, species shifts, or decreased decomposition rates.

Decomposition rates are decreased in acidified streams, resulting in reduced energy flow to detritivores, the primary base of food support for fish, and also resulting in lowered rates of remineralization and recycling of nutrients.

Invertebrates vary in sensitivity. Mayflies (Ephemeroptera) appear to be very sensitive to acid waters as a group, while stoneflies (Plecoptera) are not. Caddisflies (Trichoptera) appear to contain both sensitive and tolerant species. Snails, clams, and many crustaceans appear to be very sensitive to acidification. Based on limited field and laboratory studies, acidification to pH 4.0 will reduce invertebrate numbers by 60 to 80 percent, will result in increased drift of collector and predator species, and will result in decreased emergence of certain groups, especially the aquatic Diptera (midges, craneflies, and biting midges) and Ephemeroptera (mayflies).

Finally, preliminary results from our studies in Michigan were summarized. Our results for invertebrates are consistent with those summarized above. However, no reduction in degradation of leaf packs is yet apparent, perhaps because of the need for conditioning of the leaves before invertebrate feeding rates are expected to increase (Cummins, 1975), low winter temperature exposures to date and the short time the experiment has been in progress.

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